LEACHATE TREATMENT

R. STEGMANN*, K.-U. HEYER** AND R. COSSU°

** Institute of Waste Management, Hamburg University of Technology, Harburger Schloßstr. 36, 21079 Hamburg, Germany*
*** Consultants for Waste Management, Prof. R. Stegmann und Partner, Nartenstraße 4a, D-21079 Hamburg, Germany*
° IMAGE, Department of Hydraulic, Maritme, Environmental and Geotechnical Engineering, University of Padua, Via Loredan 20, 35131 Padova, Italy

SUMMARY: Optimum landfill operation (f.e. bioreactor landfill) with leachate and gas collection and treatment is the basis for a safe landfill with minimized emissions. Leachate treatment is essential to reduce mainly the organic and nitrogen content in the leachate. Biological processes are widely used as f.e. activated sludge plants and aerated lagoons. But the remaining values of COD and AOX are still relatively high. This was the reason to develop physical-chemical treatment steps as alternative or additional treatment methods. Many experiences with these treatment methods have been made in the past so that leachate treatment is state of the art. Although a great number of leachate treatment plants are under operation, there is not only one general solution. The kind of leachate treatment chosen should be based on the specific situation respecting the relevant regulations and costs.

1. INTRODUCTION

Sanitary landfill leachate is a highly and complex polluted wastewater. Its quality is the result of biological, chemical and physical processes in landfills combined with the specific waste composition and the landfill water regime.

With increasing leachate effluent quality standards the efforts for leachate treatment also increase. Treatment procedures must consider the relatively small flow rates and the complex leachate composition which makes it different from sewage and other kinds of waste water.

2. LEACHATE QUALITY AND QUANTITY

2.1. Leachate quality

Mainly two different phases can be identified in landfills during the anaerobic decomposition of waste: acid phase, which causes a decrease of pH in the leachate but high concentrations of

Proceedings Sardinia 2005, Tenth International Waste Management and Landfill Symposium
S. Margherita di Pula, Cagliari, Italy; 3 - 7 October 2005

organic acids and inorganic ions (for example, Cl^-, SO_4^{2-}, Ca^{2+}, Mg^{2+}, Na^+) and the methanogenic phase. Heavy metal concentrations are in general comparatively low. Leachate from the acid phase is therefore characterized by high BOD_5 values (commonly > 10.000 mg/l), high BOD_5/COD ratios (commonly > 0,7) and acidic pH values (typically 5 - 6). Further informations on the biological degradation processes can be found elsewhere (Stegmann and Spendlin, 1989).

The stable methanogenic phase (Phase IV, Fig. 1) of anaerobic degradation is characterized by a pH range from 6 to 8. At this stage, the composition of leachate is characterized by relatively low BOD values and low ratios of BOD/COD. Ammonia continues to stay at a relatively high level.

In Table 1 the ranges of leachate concentrations depending on the two main degradation phases for some relevant parameters are presented. Ehrig (1990) compiled leachate concentrations from German landfills from the seventies and eighties. It becomes obvious that the organics (COD, BOD_5, TOC) as well as AOX, SO_4, Ca, Mg, Fe, Mn, Zn and Cr are highly influenced by the acid phase resp. methanogenic phase.

Kruse (1994) investigated 33 landfills in Northern Germany, the leachate concentrations mainly derive from the late eighties and early nineties. He defined three characteristic periods according to the BOD_5/COD-ratio:

Acid phase:	$BOD_5/COD \geq 0,4$
Transient phase:	$0,4 > BOD_5/COD > 0,2$
Methanogenic phase:	$BOD_5/COD \leq 0,2$

Between the two investigations there are significant differences concerning the organic parameters. In the younger landfills (Kruse, 1994) leachate concentrations of COD, BOD_5 and TOC are lower than those determined by Ehrig (1990) some ten years before. This can be explained by developments in the technology of waste landfilling where in many younger landfills waste compaction is practised in thin layers. In addition also the waste composition may have changed (less biodegradable waste). These effects may result in a shortening of the acid phase and to an accelerated production of methane and carbon dioxid.

2.2. Leachate quantity

Leachate production is the result of precipitation, evaporation, surface runoff, infiltration, waste storage capacity etc. In many Northern German Landfills the measured leachate production rates are often between 12 and 22 % (mean value: 18 %) of the annual precipitation. Values lower than 10% are mainly from very young landfills in operation. Values for leachate production above 25% (up to 50 %) of the annual precipitation rate are from landfills where the storage capacity has been widely used.

3. LEACHATE TREATMENT

3.1. German discharge limit values

There are a few parameters which are of great importance for the kind of treatment technology that has to be applied, mainly COD and AOX, furthermore Nitrogen and BOD_5. The first two parameters require a more comprehensive treatment technology respectively a combination of different treatment methods that are described in the following.

Table 2. Limit values for the discharge of treated leachate according to German standards (51. Anhang Rahmen-AbwasserVwV, Anonymus 1996)

Parameter	**limiting concentration mg/l**
COD	200
BOD_5	20
Nitrogen, total (Summe NH_4 + NO_2 + NO_3)	70
Phosphorus, total	3
Hydrocarbons	10
Nitrite-Nitrogen	2
AOX	0,5
Mercury	0,05
Cadmium	0,1
Chromium	0,5
Chromium (VI)	0,1
Nickel	1
Lead	0,5
Copper	0,5
Zinc	2
Cyanide, easy liberatable	0,2
Sulfide	1

Table 1. Constituents in leachates from MSW landfills (after EHRIG, 1990 and KRUSE, 1994)

Parameter	Unit	Leachate from MSW landfills (EHRIG, 1990)				Leachate from MSW landfills (KRUSE, 1994)					
		Acid phase		Methanogenic phase		Acid phase		Intermediate phase		Methanogenic phase	
		Range	Medium	Range	Medium	Range	Medium	Range	Medium	Range	Medium
pH-value	-	4,5 - 7	6	7,5 - 9	8	6,2 - 7,8	7,4	6,7 - 8,3	7,5	7,0 - 8,3	7,6
COD	mg/l	6.000 - 60.000	22.000	500 - 4.500	3.000	950 - 40.000	9.500	700 - 28.000	3.400	460 - 8.300	2.500
BOD_5	mg/l	4.000 - 40.000	13.000	20 - 550	180	600 - 27.000	6.300	200 - 10.000	1.200	20 - 700	230
TOC	mg/l	1.500 - 25.000	7.000	200 - 5.000	1.300	350 - 12.000[2]	2.600[2]	300 - 1.500[2]	880[2]	150 - 1.600[2]	660[2]
AOX	µg/l	540 - 3.450	1.674	524 - 2.010	1.040	260 - 6.200	2.400	260 - 3.900	1.545	195 - 3.500	1.725
org. N[1]	mg/l	10 - 4.250	600	10 - 4.250	600						
NH_4-N[1]	mg/l	30 - 3.000	750	30 - 3.000	750	17- 1.650	740	17- 1.650	740	17- 1.650	740
TKN[1]	mg/l	40 - 3.425	1.350	40 - 3.425	1.350	250 - 2.000	920	250 - 2.000	920	250 - 2.000	920
NO_2-N[1]	mg/l	0 - 25	0,5	0 - 25 0	0,5						
NO_3-N[1]	mg/l	0,1 - 50	3	0,1 - 50	3						
SO_4	mg/l	70 - 1.750	500	10 - 420	80	35 - 925	200	20 - 230	90	25 - 2.500	240
Cl	mg/l	100 - 5.000	2.100	100 - 5.000	2.100	315 - 12.400	2.150	315 - 12.400	2.150	315 - 12.400	2.150
Na[1]	mg/l	50 - 4.000	1.350	50 - 4.000	1.350	1 - 6.800	1.150	1 - 6.800	1.150	1 - 6.800	1.150
K[1]	mg/l	10 - 2.500	1.100	10 - 2.500	1.100	170 - 1.750	880	170 - 1.750	880	170 - 1.750	880
Mg	mg/l	50 - 1.150	470	40 - 350	180	30 - 600	285	90 - 350	200	25 - 300	150
Ca	mg/l	10 - 2.500	1.200	20 - 600	60	80 - 2.300	650	40 - 310	150	50 - 1.100	200
tot. P[1]	mg/l	0,1 - 30	6	0,1 - 30	6	0,3 - 54	6,8	0,3 - 54	6,8	0,3 - 54	6,8
Cr[1]	mg/l	0,03 - 1,6	0,3	0,3 - 1,6	0,3	0,002 - 0,52	0,155	0,002 - 0,52	0,155	0,002 - 0,52	0,155
Fe	mg/l	20 - 2.100	780	3 - 280	15	3 - 500	135	2 - 120	36	4 - 125	25
Ni[1]	mg/l	0,02 - 2,05	0,2	0,02 - 2,05	0,2	0,01 - 1	0,19	0,01 - 1	0,19	0,01 - 1	0,19
Cu[1]	mg/l	0,004 - 1,4	0,08	0,004 - 1,4	0,08	0,005 - 0,56	0,09	0,005 - 0,56	0,09	0,005 - 0,56	0,09
Zn	mg/l	0,1 - 120	5	0,03 - 4	0,6	0,05 - 16	2,2	0,06 - 1,7	0,6	0,09 - 3,5	0,6
As[1]	mg/l	0,005 - 1,6	0,16	0,005 - 1,6	0,16	0,0053 - 0,11	0,0255	0,0053 - 0,11	0,0255	0,0053 - 0,11	0,0255
Cd[1]	mg/l	0,0005 - 0,14	0,006	0,0005 - 0,14	0,006	0,0007 - 0,525	0,0375	0,0007 - 0,525	0,0375	0,0007 - 0,525	0,0375
Hg[1]	mg/l	0,0002 - 0,01	0,01	0,0002 - 0,01	0,01	0,000002 - 0,025	0,0015	0,000002 - 0,025	0,0015	0,000002 - 0,025	0,0015
Pb[1]	mg/l	0,008 - 1,02	0,09	0,008 - 1,02	0,09	0,008 - 0,4	0,16	0,008 - 0,4	0,16	0,008 - 0,4	0,16

[1] parameter more or less independent from the biochemical degradation phase
[2] DOC

3.2. Biological treatment

Biological treatment is worldwide the most common practice for leachate treatment. Biological systems can be divided in anaerobic and aerobic treatment processes. Both can be realized by using different plant concepts.

In the following some of them are presented:

anaerobic biological treatment

- parts of the landfill body used as a reactor
- anaerobic filter
- anaerobic sludge bed reactor (UASB)

aerobic biological treatment

- aerated lagoons
- activated sludge plants
- rotating biological contactors (RBC)
- trickling filter
- sequential batch plant
- co-treatment with sewage

3.2.1. Biological co-treatment of sewage and landfill leachate

Data of co-treatment experiments show that this technology is a possible way to treat leachate (Kayser, 1986; Dahm, 1994). The results can be supported by the fact that worldwide sewage as well as leachate are successfully biologically treated separately (Heyer and Stegmann, 1998). From this it can be concluded that also combinations of these wastewaters can be treated biologically. But it is necessary to respect some consequences when leachate is added to a biological sewage treatment plant. In this case the organic loading increases, where it is necessary not to overload the sewage treatment plant. Special attention has to be brought to the high ammonia concentrations in the leachate also from landfills in the methanogenic phase. If the loading is in correspondence to the design values of the sewage treatment plant no increase of nitrogen- or BOD_5 concentrations in the effluent are expected. A restriction may arise when sewage is used as a carbon source for denitrification of nitrate which originates from a mixture of sewage and leachate, where the latter comes from old landfills with low organic degradable concentrations.

The increase of nondegradable organic leachate components (especially residual COD, AOX) is mainly a function of dilution. Problems may arise when the COD in the effluent of the sewage treatment plant exceeds the limit values due to the addition of leachate. In this case higher costs may arise for the plant operator, because he may has to pay for the COD-load that is above the discharge limit.

Since the concentrations of heavy metals are in general also in non-treated leachate relatively low, negative effects from the co-treatment of sewage and leachate are not expected. An exception may be leachate from landfills in the acetic phase where zinc concentrations may be elevated (see also Table 1). During biological treatment most of the zinc precipitates and remains in the sludge. There may be differences in leachate concentrations from other landfills where the landfilled waste composition and / or other conditions are different.

Overall, biological co-treatment of sewage and leachate is a proven technology and operates in general well, if the treatment plant is carefully designed and operated and not overloaded. Experiences have shown that during the co-treatment of plants sewage and leachate special emphasis has to put on the ratio of BOD_5 to nitrogen. In addition it has to be proven that the leachate does not contain toxic substances. The acceptance of this leachate treatment procedure varies widely beyond sewage treatment plant operators. In general biological degredation tests in the laboratory are recommended, respecting the specific situation.

3.2.2. Anaerobic treatment

During the period of high organic concentrations in leachates from the acidogenic phase of a landfill (Table 1) an anaerobic treatment step might be a way of reducing main proportions of the degradable organics (Mennerich, 1988). The main advantage of the anaerobic treatment process is the low energy requirement, because no oxygen has to be supplied. Technical anaerobic processes need adaequate temperatures of 35° C resp. 55° C. The process is very sensible in regard to varying milieu conditions. Anaerobic leachate treatment is an effective process but the remaining BOD_5- und COD-effluent concentrations are still high with COD-values of 1.000-4.000 mg/l and a BOD_5 / COD-ratios > 0.3. After the anaerobic treatment step the leachate has to be treated to final effluent standards by means of aerobic processes. During investigations and under operational conditions anaerobic filters were sometimes clogged due to iron and calcium precipitation. The free volume of the reactors was in one case consumed up to 60 % by the precipitates after a COD-reduction of 2.000-3.000 kg/m^3. At UASB reactors the anorganic content of sludge increases dramatically with time and reduces elimination rates. (Mennerich, 1988)

Only under the condition that huge landfills are operated over long periods of time an anaerobic treatment step may be considered. In general the authors do not recommend anaerobic treatment due to the fact that the leachate quality (esp. the BOD) changes after some years, when the methanogenic phase starts. When the BOD_5 concentrations are < 2.000 – 5.000 mg/l anaerobic processes are not feasible.

If a landfill consists of old and young parts than also the leachate from the young part (which is in the acidogenic phase) can be recirculated on the old part (which is in the methanogenic phase) where the high organic acids from the leachate are degraded and the leachate changes from “acidogenic” to “methanogenic” leachate.

Another mechanism to reduce the BOD in the leachate also from “young” landfills is a 1-2 m layer of composted waste which is placed on the surface of the drainage system which is installed above the bottom liner. This layer acts as an anaerobic filter and reduces the organic acid concentrations significantly (Stegmann and Spendlin, 1989).

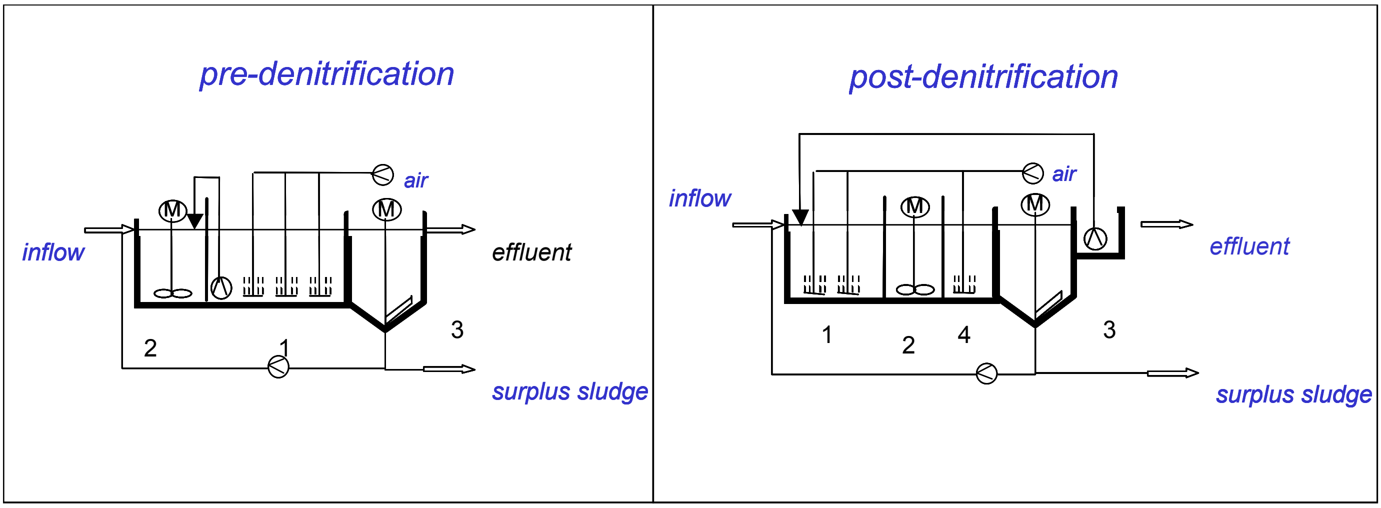
3.2.3. Aerated lagoons

Aerated lagoons are a relatively simple leachate treatment system. The basic idea is that the retention time of the leachate is long enough so that as many bacteria can develop per time as the number that has been transported out of the lagoon with the effluent. Long retention times are also necessary in order to oxidise ammonia nitrification especially during low temperatures. The maintenance and operation costs are relatively low. The detention times that are necessary are in the range of 50 – 100 days (Cossu et. al. 1989).

3.2.4. Activated sludge plants

The detention time in activated sludge plants can be considerably shorter than in aerated lagoons. The reason is that the sludge content (amount of bacteria) can be controlled which is several times higher than in aerated lagoons. This is achieved by installing a settling tank behind the aeration tank and recirculating the sludge back into the activated sludge tank. A certain amount of sludge has to be removed as excess sludge out of the system.

Beside BOD_5-reduction the nitrification of ammonium is necessary.



1. nitification tank; 2. denitrification tank; 3. settling tank; 4. aeration tank

Figure 2. Schemes of nitrification / denitrification plants

Nitrogen elimination becomes more and more important with the age of a landfill and the increasing reduction of BOD in the landfill leachate. The treatment of such a leachate is more complicated than the one with a high BOD_5 content.

The pH of these leachates is mainly in the area of 8.0 – 8.5. But during aeration the pH increases in some cases up to pH 9 and higher. Under these circumstances the equilibrium shifts from ammonium to free ammonia in the gas phase. Concentrations of free ammonia may have an inhibiting effect on nitrifying bacteria. If on the other side the ammonium is converted to nitrate the pH decreases as a result of alcalinity consumption. Overall a very careful operation and pH-control is necessary to get low ammonium effluent values.

In order to prevent low temperatures in the activated sludge plants it may be necessary to cover the aeration tank and to use part of the heat from the air supply blowers when bubble aeration is used for heating.

To reduce the high nitrate content in the leachate effluent and to stabilize pH-conditions in activated sludge plants a denitrification step is necessary. The pre-denitrification would be more effective if there are adaequate high concentrations of degradable organics in the raw leachate, that can be used as a substrate for the denitrifying bacteria.

The denitrification rate depends on the amount of recirculated water and sludge rates and BOD_5 to N-ratio (see Fig. 1) To reach nitrogen effluent values lower than 5 to 10 % of the influent concentrations extremely high recirculation rates are necessary.

Using a post-denitrification process very low nitrate effluent values are possible. But organics in the leachate cannot be used as a carbon source and a separate pH-stabilization for the nitrification tank may be necessary. Leachate from the methanogenic phase can only be denitrified with the addition of an external carbon source as f.e. acetic acid, methanol etc. (see Fig. 2).

3.2.5. Rotating biological contactors (RBC) and trickling filters

These plants differ from the activated sludge plants in so far that the bacteria are attached to the material of the rotating contactors or the fillings in the trickling filters. The air supply takes place naturally, i.e. the rotating contactor is partly in the air and partly in the water. Air may vent naturally or artificially through a trickling filter. This treatment method consumes relatively low amounts of energy. Treating high organic polluted leachates may result in a clogging by means of anorganic precipitates and/or produced biomass. On the other hand in many cases nitrification processes are more effective in fixed film reactors due to the high sludge age. For this reason this treatment methods are more appropriate for the treatment of leachate from old landfills. The influence of temperature on nitrification is relatively strong. Temperature effects are easier to control at RBC's, because they are compact and normally covered.

3.2.6. Summary of biological treatment

Biological treatment processes are very effective methods to reduce biodegradable organics as BOD_5 and the main part of COD. Also from leachates with low organic concentrations and BOD_5 / COD-ratio < 0.2 the COD may be removed by biological treatment up to 50 %. It is also an effective method to oxidize ammonium to nitrate and to reduce nitrate by means of denitrification to gaseous nitrogen. The decreasing elimination rates during periods of low water temperatures especially for ammonium reduction are a disadvantage.

Since the concentrations of phosphorous in the leachate are in general too low. When biological leachate treatment processes are used, phosphorous has to be added in most cases. Adaequate P-dosing is necessary in order to install the appropriate BOD – N – P-ratio. In addition problems like intensive foam production in the activated sludge basin may occur. High precipitation rates of f.e. iron - and carbonate compounds have to be expected so that on a routine base cleaning of submersial pumps, aeration devices etc. is necessary.

Using only biological leachate treatment the COD and propabely AOX-concentrations will not meet f.e. the German effluent standards. For this reason further treatment ist necessary.

4. CHEMICAL OXIDATION

During the last years chemical oxidation processes have been resp. still are used at different landfills in Germany. A combination of oxidation agents as ozone or hydrogen peroxide and ultraviolet light (UV) is often used. This combination shows high oxidation rates for COD and AOX in the leachate. The process consists of a chamber to mix influent leachate and the oxidation agents. In most cases this process is supported by means of UV-treatment. To increase elimation rates manifold of leachate volumes are recirculated. In opposite to mixing hydrogen peroxide and water the mixing of gaseous ozone and water is more difficult. One main soal is a maximum utilisation of the ozone for oxidation. Often the ozone utilisation rates are too low which makes the process costly.

Figure 3 shows as an example of a treatment plant in Northern Germany a process sequence for biological treatment in combination with a chemical oxidation (ozone). Average concentrations for COD, ammonium and AOX of the influent, the effluent of the biological and of the chemical oxidation as well as of the second biological treatment step in the rotating biological contactor are presented.

It has to be encountered that also anorganic compounds may be oxidised during the chemical oxidation step. To prevent the expensive oxidation of easy biodegradable components a biological pre-treatment including nitrification / denitrification should be considered.

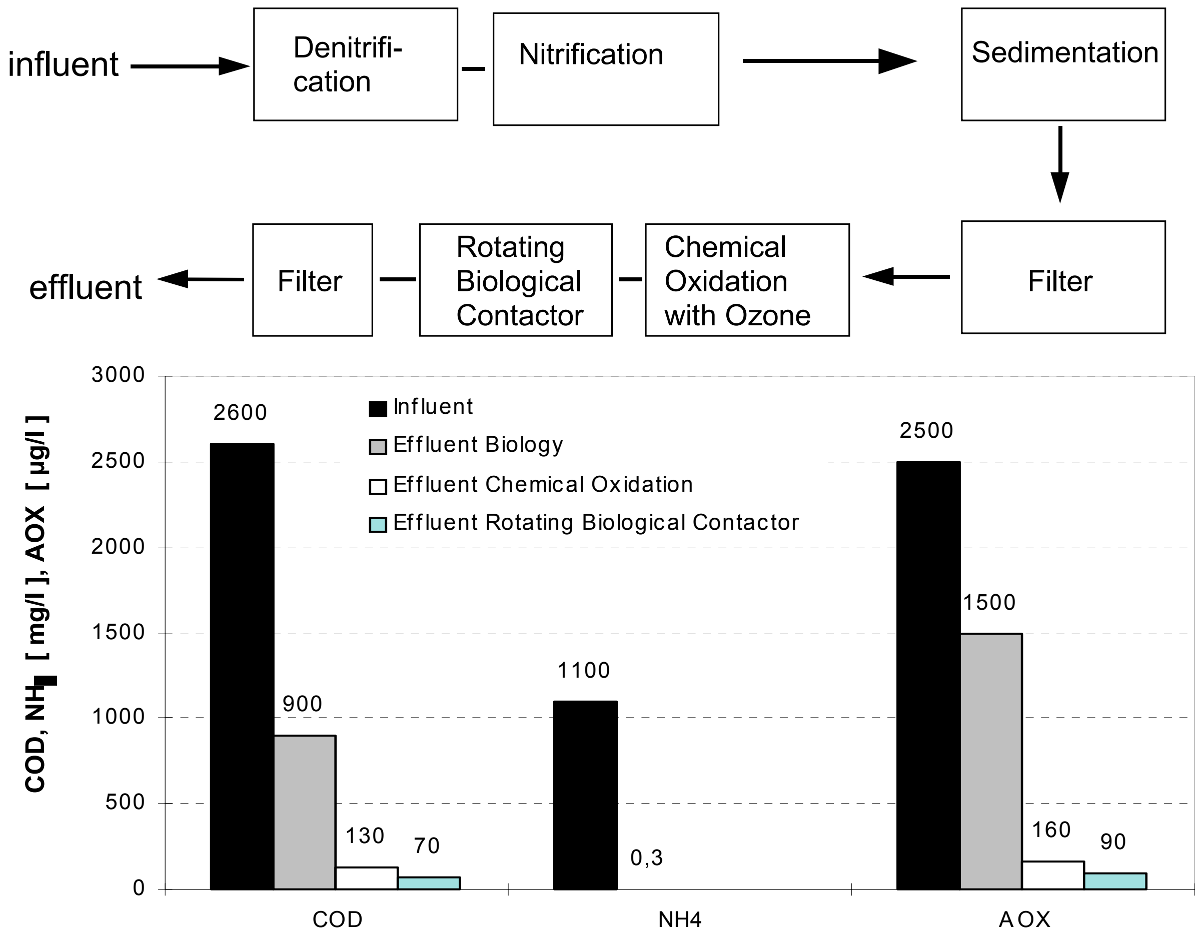


Figure 3. Processing sequence for a combination *biology* + *chemical oxidation* + *biology*; characteristic leachate concentrations of a treatment plant (adopted from ATV 7.2.26, Anonymus 1996)

During chemical oxidation not all organics are oxidized to carbon dioxide and water. Some organics are only partly oxidized often to biological degradable intermediale products. These “new” biodegradable organics shall be reduced by biological treatment. For the reduction of these relatively low concentrations of organics a fixed film reactor may be an option. It can also be considered to feed the effluent of the chemical oxidation plant back to the influent of the biological reactor.

4.1. Activated carbon

In Germany for the reduction of the residual COD- and AOX-concentrations below the limit discharge values more and more activated carbon is used. This is due to the fact that the costs for activated carbon have decreased in the recent years in Germany significantly.

Two different systems may be used:

- Addition of powdered activated carbon into a reaction tank (mainly with biologically preatreated leachate). The loaded activated carbon is removed by means of precipitation / floccutation processes using iron or aluminum salts resp. organic flocculants. The precipitated activated carbon is removed by means of a settling tank following the reaction tank. This technic was used quite often predominantly in the past.

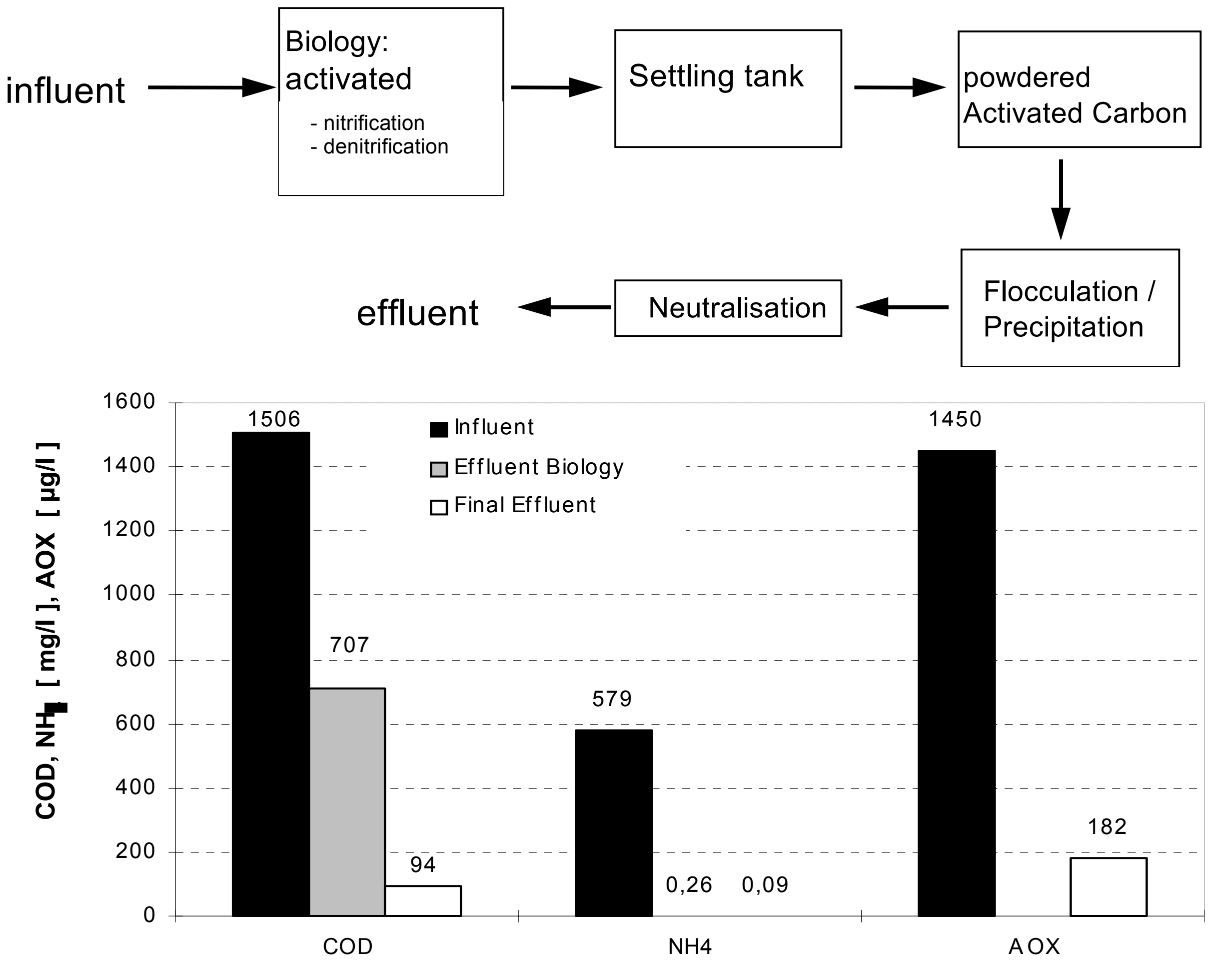


Figure 4. Processing sequence for a combination *biology* + *adsorption (activated carbon)* + *flocculation/precipitation*; characteristic leachate concentrations of a treatment plant (adopted from ATV 7.2.26, Anonymus 1996)

Nowadays more granular carbon is used, which is placed in columns, where the leachate migrates through. The advantage of this process is the avoidance of the flocculation precipitation step in order to remove the powdered activated cabon, which in general also results in an increase in the salt content. In addition it is a great advantage that a re-utilisation of the loaded granular carbon after it has been thermally regenerated is common practice.

In Figure 4 as an example a process sequence for biological treatment in combination with adsorption (powdered activated carbon) and a flocculation/precipitation is presented. Average concentrations for COD, ammonium and AOX of the influent, the effluent of the biological and of the final effluent are presented in Fig. 4.

4.2. Flocculation / Precipitation

Flocculation/Precipitation e.g. with $FeCl_3$, is mainly practised to reduce the organic load (humic acids and halogenated organic constituents characterized by the parameters COD and AOX) of the leachate after the biological treatment. This technology is not used frequently also due to the fact of the increase of chloride and/or sulfate in the leachate effluent. If powered activated carbon is used, a flocculation / precipitation step will be necessary for the removal of the loaded activated carbon.

4.3. Physical-chemical processes

4.3.1. Reverse Osmosis

One of the developments in the last decade for leachate treatment is the reverse osmosis (RO). But in contrast to the biological treatment it is a separation process into two streams - one low polluted permeate stream and one highly polluted concentrate stream. Using this technology it is possible to produce very low concentrated permeates. If leachate from the acetic phase has to be treated a biological pretreatment may be necessary for several reasons as increased precipitation has to be expected, low molecules may pass the membrane and fowling on the membrane surface may be enhanced. During reverse osmosis the separation of ammonium is often not sufficient. The reduction of ammonia concentrations in the permeate may be increased by menas of a two or multiple step reverse osmosis. In some cases ammonium is removed by means of a pre-stripping process or a biological nitrification and denitrification step.

A disadvantage of RO is the production of the liquid concentrate (about ± 20 % of the leachate). The technique of back passing the concentrate into the landfill is in the opinion of the authors not the best option. At present in Germany evaporation of the concentrate is used at a few landfills. Other ways of concentrate disposal are beyond others solidification and (deep mine) landfilling, incineration in hazardous waste or municipal solid waste incinerators.

Figure 5 shows a process sequence for biological treatment in combination with a reverse osmosis at a landfill in Northern Germany. Concentrations for COD, ammonium and AOX of the influent, the effluent of the biological and of the reverse osmosis are also presented.

Several of these combination plants are in operation in Germany. If leachate from the methanogenic phase has to be treated, the biological treatment step may be deleted. The ammonia has to be reduced in this case by means of stripping or (mulitiple step) reverse osmosis. The latter procedure is used more often.

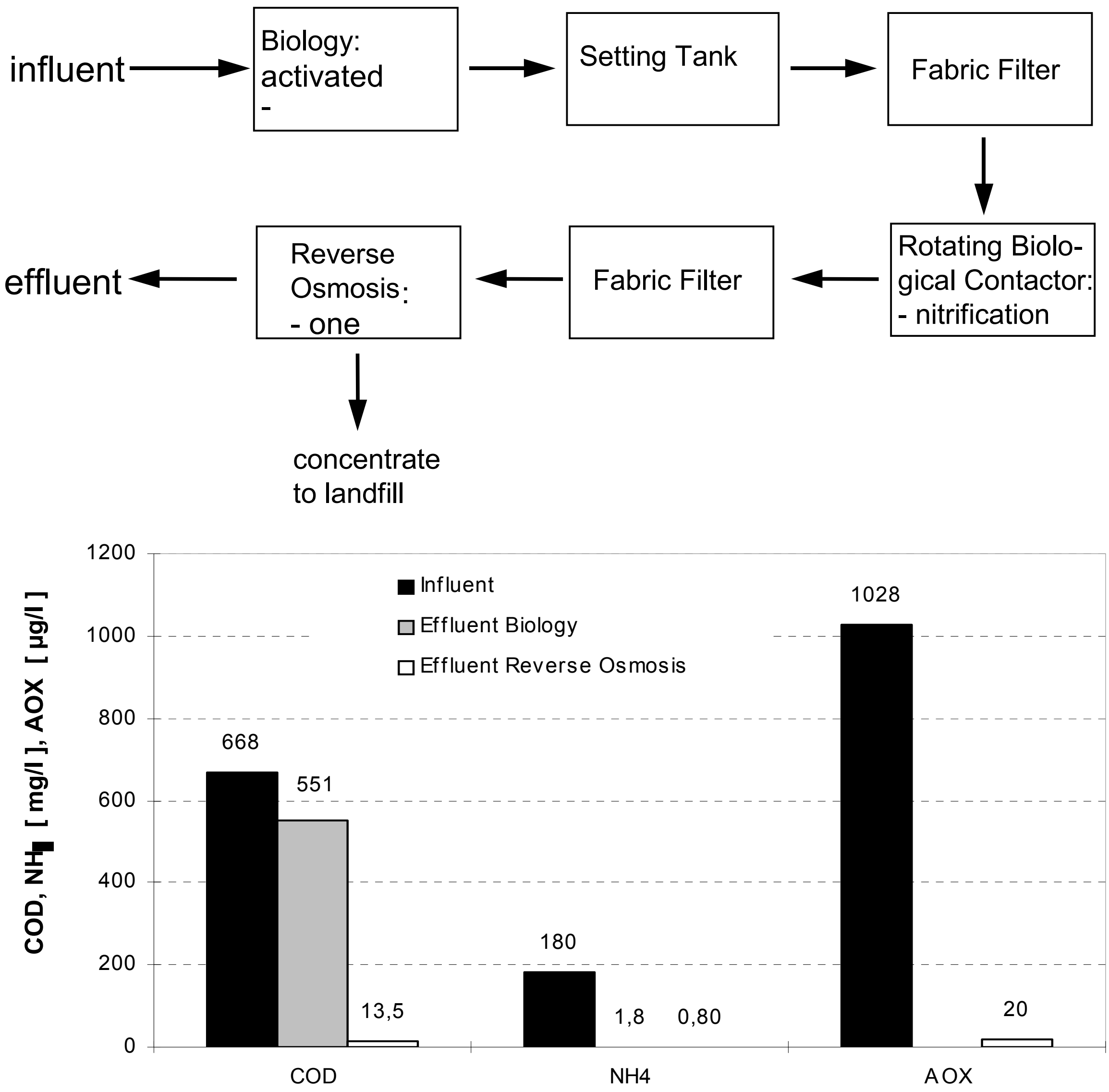
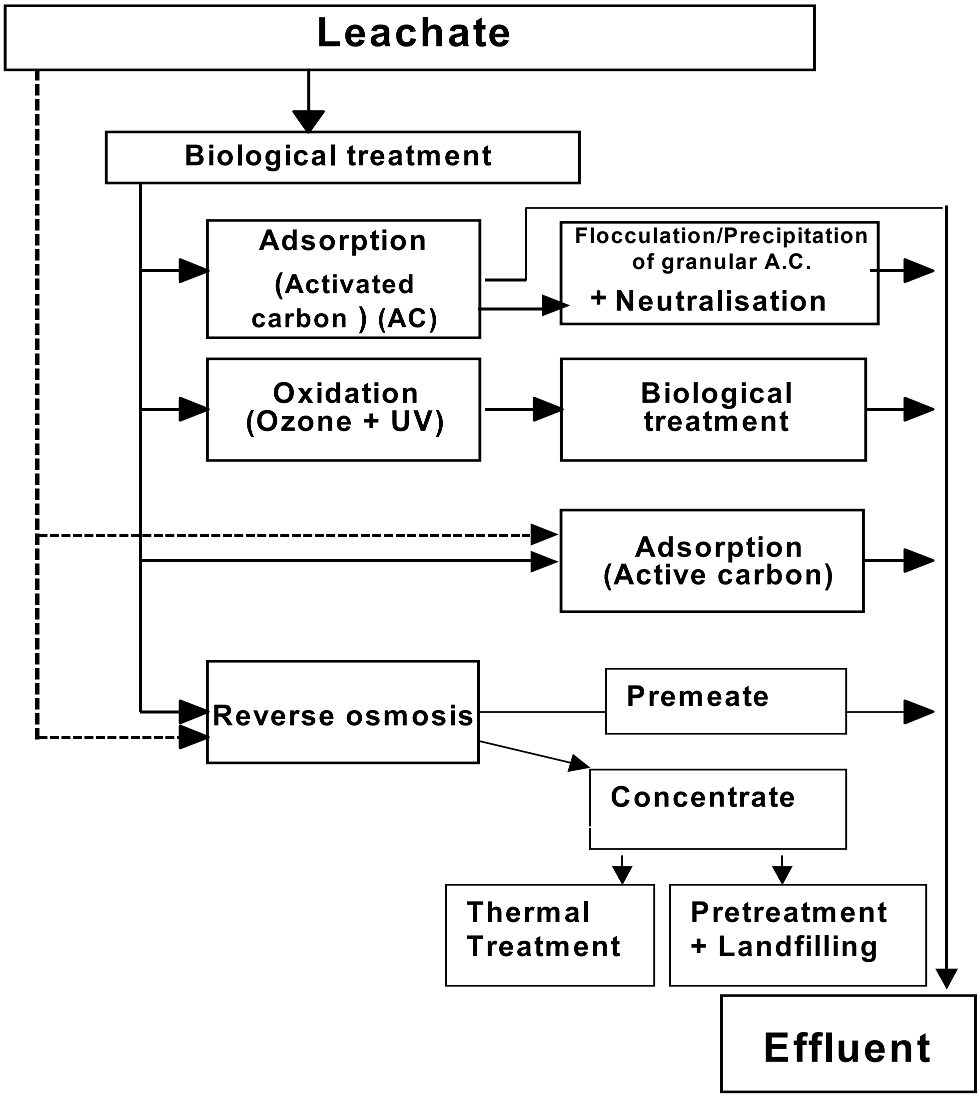


Figure 5. Processing sequence for a combination *biology* + *reverse osmosis*; characteristic leachate concentrations of a treatment plant (adopted from ATV 7.2.26, Anonymus 1996)

4.4. Combination of treatment methods

Plants currently in operation for the treatment of leachate often consist of several of the above mentioned treatment methods to meet the limiting concentrations for the effluent. Leachate from landfills in the methanogenic stage may also be treated by a single step (f.e. multi-step reverse osmosis). Typical combinations are shown in Figure 6.



-------- : potential treatment for leachate from the methanogenic phase of a landfill
AC: Activated Carbon
UV: ultraviolet light

Figure 6. Schemes of often used methods and combinations for leachate treatment (modified by Ehrig et al., 1998)

For some combinations average influent and effluent concentrations of characteristic leachate components are presented in Table 3. All the presented treatment combinations may reach the German discharge limit criteria. These plants have been operated for several years and can be regarded as state of the art treatment facilities.

Table 3. Examples of influent and effluent concentrations of different combinations for leachate treatment (ATV Group 7.2.26, Anonymus, 1993)

COD [mg/l] Influent	COD [mg/l] Effluent	NH_4-N [mg/l] Influent	NH_4-N [mg/l] Effluent	AOX [mg/l] Influent	AOX [mg/l] Effluent
A: biology - active carbon - flocculation/precipitation - neutralisation					
1571	76	579	0,09	1,45	0,18
686	51	528	0,7	1,65	0,23
B: biology - ultrafiltration - activated carbon					
1000 - 12000	< 200	400 - 800	< 10	1 - 2,5	0,1 - 0,7
C: biology - chemical oxidation (ozone + UV)					
320 - 5796	30 - 137	125 - 1350	0,4 - 36,2	-	-
D: biology - chemical oxidation (ozone + UV) - biology					
1200 - 4000	18 - 150	600 - 1900	0,1 - 9	1 - 3,8	0,04 - 0,18
E: biology - chemical oxidation (ozone + UV) - biology - activated carbon					
758 - 1332	1 - 85	375 - 885	0,1 - 0,6	0,85 - 2,1	0,17 - 0,43
F: one step reverse osmosis					
4124	20	577	8	-	-
1550	68	750	7	1,4	< 0,01
G: two step reverse osmosis					
1590 - 2980	4 - 25	900 - 1800	4,4 - 8,8	1,5 - 1,9	0,002 - 0,02
H: biology - reverse osmosis					
446 - 872	5,3 - 27	80 - 396	0,03 - 10,1	0,4 - 1,4	< 0,01 - 0,05
I: biology - two step reverse osmosis					
1366 - 3010	< 2	130 - 854	6,3	1,09 - 2,24	0,045

4.5. Costs of leachate treatment

The costs of leachate treatment in Germany can be only roughly estimated because they vary roughly between 10 and 70 €/m^3 leachate

This has several reasons:

- the same treatment procedures may be totally different put into practice; treatment facilities may be installed in cheap containers or in expensive buildings
- the technical equipment can be very simple or very sophisticated, e.g. for on-line measurements of the leachate components
- the total capacity and the utilization coefficient of the treatment plant; a small capacity and a low utilization coefficient means high costs per m^3 of treated leachate
- a growing competition between companies who are producing treatment plants leads to lower prices
- decreased prices for energy and chemicals like oxygen or active carbon
- cut backs in the budgets of landfill operators for landfill operation in general (reduced waste quantities for disposal, dropping prices for waste, growing competition between landfill operators etc.)

In 1994 seven treatment plants with capacities from 11.000 up to 64.000 m^3/a were investigated. The total costs for investment and operation varied between 9 € and 30 €/m^3 leachate.

5. CONCLUSIONS

The presented paper shows different methods for leachate treatment. Leachate control is a very important step to receive the long-term functionality of the drainage system, to reduce treatment costs and to render possible high-tech treatment systems. Nowadays more than 100 leachete treatment plants are under operation in Germany, so there are many experiences concerning the technology, costs, the effluent quality and associated problems. In some cases the treatment of leachate resulted in increasing operation problems in opposite to the treatment of other wastewaters. The selection of the adaequate treatment process should not only include the compliance with the effluent limit values and maintenance but also the production of residuals which have to be further treated or disposed.

The decrease of discharge limits values in many countries in the past and in future requires high-tech treatment technologies and often a combination of different processes. For economically developing countries simple systems as lagoons and/or constructed wetlands may be a first approach in order to decrease the main balk of the pollutants in the leachate.

REFERENCES

ANONYMUS (1993): Technical Instructions on Waste from Human Settlements (TA Siedlungsabfall). Dritte Allgemeine Verwaltungsvorschrift zum Abfallgesetz vom 14. Mai 1993, Technische Anleitung zur Verwertung, Behandlung und sonstigen Entsorgung von Siedlungsabfällen, Bundesanzeiger Nr. 99a

ANONYMUS (1996): Anhang 51: Oberirdische Ablagerung von Abfällen. Allgemeine Rahmen-Verwaltungsvorschrift über Mindestanforderungen an das Einleiten von Abwasser in Gewässer, German regulation

CAZIANI, R., COSSU, R. (1989): Landfill Hydrology and Leachate Production, from Chrisensen, Cossu, Stegmann (Eds.) Sanitary Landfilling: Process, Technology and Environmental Impact, Acadamic Press.

CHRISTENSEN, T.H., KJELDSON, P. (1989): Basic Biochemical Processes in Landfills, from: Chrisensen, Cossu, Stegmann (Eds.) Sanitary Landfilling: Process, Technology and Environmental Impact, Acadamic Press.

COSSU, R., STEGMANN, R., ANDREATTOLA, G., CANNAS, P. (1989): Biological Treatment, from: Chrisensen, Cossu, Stegmann (Eds.) Sanitary Landfilling: Process, Technology and Environmental Impact, Acadamic Press.

DAHM, W., KOLLBACH, ST., GEBEL, J. (1994): Sickerwasserreinigung: Stand der Technik 1993/94; zukünftige Entwicklungen, EF-Verlag für Energie- und Umwelttechnik, ISBN 3-924511-79-9

EHRIG H.-J. (1990): Beitrag zum quantitativen und qualitativen Wasserhaushalt von Mülldeponien, Veröffentlichungen des Institut für Stadtbauwesen, TU Braunschweig, 2nd edition

EHRIG, H.-J., HAGEDORN, S. (1998): Sickerwasserbehandlung, Belastungen, Verfahren, Entwicklung und Bewertung. in: Entwicklungstendezen in der Deponietechnik, 1. Hamburger Abfallwirtschaftstage 28.-29. Januar 1998, Hamburger Berichte, Band 12, Economica Verlag Bonn

HEYER, K.-U., STEGMANN, R. (1998): Landfill Systems, Sanitary Landfilling of Solid Wastes, Long-Term Problems with Leachates. in: Biotechnology, Vol. 11a, Wiley-VCH Verlag

KAYSER R. (1986): Leistungsfähigkeit kommunaler Kläranlagen für die Sicker-wasserbehandlung, Proceedings: Deponiesickerwasserbehandlung, Technische Universität Aachen, Germany

KRUSE, K. (1994): Langfristiges Emissionsgeschehen von Siedlungsabfalldeponien, Veröffentlichungen des Instituts für Stadtbauwessen, Technische Universität Braunschweig, Heft 54

MENNERICH, A., (1988): Beitrag zur anaerob-aeroben Behandlung von Sickerwässern aus Hausmülldeponien, Veröffentlichungen Institut für Siedlungswasserwirtschaft TU Braunschweig, Heft 44

STEGMANN, R., SPENDLIN, H.H. (1989): Enhancement of degradation: German experiences, in: *Sanitary Landfilling: Process, Technology and Environmental Impact*, (CHRISTENSEN, T.H.,COSSU, R., STEGMANN, R. Eds.), Academic Press, London UK, pp. 61 - 82.